# Covering over the cracks in conservation assessments at EU interfaces: A cross-jurisdictional ecoregion scale approach using the Eurasian otter (Lutra lutra)

Post-Print (Ecological Indicators)

**Covering over the cracks in conservation assessments at EU interfaces: A cross-jurisdictional ecoregion scale approach using the Eurasian otter (*Lutra lutra*)**

Neil Reid[a,*], Mathieu G. Lundy[b], Brian Hayden[a,c], Tony Waterman[d], Declan Looney[d], Deirdre Lynn[e], Ferdia Marnell[e], Robbie A. McDonald[f], W. Ian Montgomery[a,g]

[a] *Quercus, School of Biological Sciences, Queen's University Belfast, Belfast BT9 7BL, UK*

[b] *Agri-Food and Biosciences Institute (AFBI), Fisheries and Aquatic Ecosystems Branch, Newforge Lane, Belfast BT9 5PX, UK*

[c] *University of Helsinki, Kilpisjärvi Biological Station, Faculty of Biological and Environmental Sciences, Viikinkaari 9, Helsinki, Finland*

[d] *Northern Ireland Environment Agency (NIEA), Biodiversity Unit, Klondyke Building, Gasworks Business Park, Lower Ormeau Road, Belfast BT7 2JA, UK*

[e] *National Parks & Wildlife Service (NPWS), Department of Arts, Heritage and the Gaeltacht, 7 Ely Place, Dublin 2, Ireland*

[f] *Environment and Sustainability Institute, University of Exeter, Penryn Campus, Penryn, Cornwall TR10 9EZ, UK*

[g] *School of Biological Sciences, Queen's University Belfast, Belfast BT9 7BL, UK*

∗ Corresponding author at: *Quercus*, School of Biological Sciences, Queen's University Belfast, Belfast BT9 7BL, Northern Ireland (UK). Tel.: +44 28 9097 2281.

E-mail address: neil.reid@qub.ac.uk (N. Reid).

*Keywords:* Distribution; EC Habitats Directive; Trans-boundary; Population estimation; Range; Spraint surveys.

## ABSTRACT

Throughout the European Union, the EC Habitats Directive requires that member states undertake national surveillance of designated species. Despite biological connections between-populations across- borders, national assessments need not be co-ordinated in any way. We conducted a trans-boundary assessment of the status of Eurasian otters (*Lutra lutra*) aimed at providing consistency across a single biogeographical unit, i.e. the island of Ireland, comprising two states, i.e. the Republic of Ireland and the United Kingdom (Northern Ireland). Our aim was to ensure consistency with previous assessments conducted separately in each state, and permit each Government to fulfil their separate statutory reporting commitments. The species range increased by 23% from 1996–2006 and 2007–11. The population estimate of 9400 [95%CI 8700–12,200] breeding females during 2010/11 was not significantly different from 8300 [95%CI 7600–9800] breeding females established as a baseline during 1981–82. Modelling of species-habitat associations suggested that available habitat was not limiting and no putative pressures recorded at sites surveyed negatively affected species occurrence. Thus, under the statutory parameters for assessing a species' conservation status, i.e. range, population, habitat and future prospects, the otter was judged to be in 'Favourable' status throughout Ireland and in both discrete political jurisdictions. Thus, we provide a trans-boundary test case for EU member states that share habitats and species across ecoregions, ensuring conservation assessment data are standardised, synchronised, spatially consistent and, therefore, biologically relevant without compromising legal and administrative autonomy within separate jurisdictions.

## 1. Introduction

International, coordinated, monitoring programmes for species of conservation concern are essential in creating population measures for tracking changes in biodiversity, particularly where global factors interact with regional habitat loss and fragmentation (Pereira and Cooper, 2006; Schmeller, 2008; Henle et al., 2013). The EC Directive on the Conservation of Natural Habitats and of Wild Fauna and Flora (92/43/EEC), hereafter referred to as the Habitats Directive, requires that EU member states conduct surveillance of the conservation status of natural habitats and species (Article 11) and to report on the implementation of the Directive, including surveillance, every 6 years (Article 17). Each sovereign nation reports separately to the European Commission despite sharing habitats, species and often common borders with their neighbouring countries. Moreover, despite guidance on common standards monitoring, states can adopt various approaches to monitoring: surveys need not be temporally synchronised, survey effort can be spatially heterogeneous and different sources of data can be used, leading to spatiotemporal discontinuities in the quantity and quality of assessments. Clearly, such a discrete and jurisdictional approach could lead to biogeographical regions shared by member states drawing on inconsistent or incomplete data in making their assessments due to: divergent ecological trends within different areas; or, application of differing methods and/or interpretation of monitoring results. Thus, international conservation assessments of species and habitats could benefit from standardised surveillance throughout the range of target species or habitats (Schmeller, 2008).

The Eurasian otter (*Lutra lutra* L. 1758) is listed in Annex II of the Habitats Directive and as such requires detailed surveillance. Otters underwent a dramatic decline throughout Europe during the 20th century (Mason and MacDonald, 1986), linked principally to the bioaccumulation of polychlorinated biphenyl (PCB) pesticides (Gutleb and Kranz, 1998; Mason and Wren, 2001) but also declines in water quality, changes to watercourses, landscape intensification and invasive species (Kruuk, 1995; Chanin, 2013). The otter is currently listed by the International Union for the Conservation of Nature (IUCN) as ‘near threatened’ (Ruiz-Olmo et al., 2008). Otters are elusive and nocturnal, thus monitoring uses species inci- dence data derived indirectly from field signs. Otter surveillance started during the late 1970s and so pre-dates the Habitats Directive (West, 1975; Macdonald and Mason, 1976; Lenton et al., 1980).

The Standard Otter Survey method developed by Lenton (1980) is typically comprised of a search of 600 m of river bank for otter spraint. Changes in the frequency of positive sites have often been taken as an indication of changes in abundance (Jefferies, 1986). Consequently, this approach to otter surveillance has been adopted by most European countries as the basis of their reporting commitments. Recent research has shown that the Standard Otter Survey method is highly biased by surveyor experience and search effort, the volume of rainfall in the week before survey and the number of bridges present on the survey stretch of river calling into question its validity in determining temporal trends in abundance (Parry et al., 2013; Reid et al., 2013a). Specifically, spatial or temporal trends within- or between-countries could be entirely attributable to varying sources of survey bias.

The previous Article 17 conservation assessment for otters in the Republic of Ireland was deemed as unfavourable inadequate U1 or poor (NPWS, 2008), principally due to a decline in species incidence from 92.5% (Chapman and Chapman, 1982) to 70.5% (Bailey and Rochford, 2006) translating into a 24% decline in estimated numbers from 8400 to 6400 adult breeding females (Marnell et al., 2011). In contrast, data for Northern Ireland was reported under a submission covering the United Kingdom, which judged the otter as favourable or good (JNCC, 2007) due to a 527% increase in species incidence in England and a 268% increase in Wales (due to recent recolonization after local extirpation) with concomitant increases in overall estimated abundance despite an apparent decline from 72.4% (Chapman and Chapman, 1982) to 62.5% (Preston et al., 2006) in site occupancy throughout Northern Ireland. Great Britain and Northern Ireland, whilst forming the United Kingdom, represent distinct biogeographical ecoregions and thus the ecological relevance of changes in Northern Ireland otter numbers was lost by regional inclusion with Great Britain. Moreover, no formal comparative assessment of temporal trends in otter status has been made between Northern Ireland and the Republic of Ireland even though they are more comparable and more ecologically relevant to one another.

We aimed to conduct a conservation assessment for the otter, using Habitats Directive parameters, throughout the island of Ireland treating it as a single ecoregion.

## 2. Methods

A total of 1229 survey sites were selected throughout Ireland (853 in the Republic of Ireland and 377 in Northern Ireland) from three key habitats: flowing freshwater, representing rivers, streams and canals (n = 999), static freshwater representing lakes and reservoirs (n = 59) and coastal sites (n = 171). Sites included 525 sites surveyed previously in the Republic of Ireland (Bailey and Rochford, 2006) and 377 sites surveyed previously in Northern Ireland (Preston et al., 2006). An additional 327 sites were added to fill in gaps in the distribution including a greater proportion of coastal sites which were under-represented in previous surveys. New sites were situated on separate rivers at least 5 km apart and from existing survey sites to provide spatial independence. Sites in the Republic of Ireland were surveyed by 75 conservation rangers from the National Parks & Wildlife Service (NPWS), whilst sites in Northern Ireland were surveyed by two ecologists from Queen’s University Belfast. Surveyor training courses were held to standardise data collection whilst providing a demonstration of the survey protocol in the field. Surveys were based on the ‘Standard Otter Survey’ method (Lenton et al., 1980) in which a 600 m stretch of river was selected and one bank walked searching exposed boulders, bridge footings and the bankside for otter spraint. The method was modified at lakes and the coast to include 600 m of shoreline.

Methods for assessing conservation status have been devised by the European Topic Centre for Nature Conservation (ETCNC) in conjunction with EU member states represented on the Scientific Working Group of the Habitats Directive (Evans and Arvela, 2011). The conservation status of a species is assessed on four parameters scored objectively: i) range; ii) population; iii) habitat; and iv) future prospects. Conservation status is defined as "the sum of the influences acting on the species concerned that may affect the long- term distribution and abundance of its populations". A standard format for reporting, but not data collection, was agreed at a European level during 2006 (European Commission, 2006). The format involves the application of a traffic-light system and brings together information on the four parameters to be assessed. Each parameter is classified as being 'favourable FV' or 'good' (green), 'unfavourable inadequate U1' or 'poor' (amber), 'unfavourable U2' or 'bad' (red) and 'unknown' (grey). A species is taken as favourable only when: i) population dynamics data on the species concerned indicate that it is maintaining itself on a long-term basis as a viable component of its natural habitats; ii) the natural range of the species is neither being reduced nor is likely to be reduced for the foreseeable future; and, iii) there is, and will probably continue to be, a sufficiently large habitat to maintain its populations on a long-term basis. Favourable reference values for range and population are set as targets against which future values can be judged. These reference values have to be at least equal to the value when the Habitats Directive came into force i.e. in 1994. The 'Favourable Reference Range' for a species is the geographic range within which it occurs and which is sufficiently large to allow its long-term persistence. The major pressures and threats perceived to be affecting the species are listed during each assessment. Their status, projected status and observed impacts are used to determine the species' likely future prospects. If any one of the four parameters are assessed as unfavourable, then the overall assessment for the species is also unfavourable.

*2.1. Range*

The previous Article 17 report under the Habitats Directive established a baseline Favourable Reference Range for the otter between the implementation of the Directive and the submission of the first report covering the 13 year period from 1993 to 2006. The species range was described at a 10-km square scale consistent with methods adopted by species atlases. The Directive requires reporting every 6 years constraining the period during which the current distribution (i.e. occupied 10-km grid cells) could be assessed, that is, the 4 year period from 2007 to 2011. This necessarily constrained the methodology that could be employed to describe changes in the distribution of the otter. Species records from the current survey were augmented with those from multiple sources including the Centre for Environmental Data and Recording (CEDaR), the National Biodiversity Data Centre (NBDC), the Environmental Protection Agency (EPA), National Parks and Wildlife Service (NPWS), Northern Ireland Environment Agency (NIEA), members of the public and www.biology.ie (courtesy of Paul Whelan). Otter distribution during 2007–2011 was compared to that recorded at baseline during 1993–2006 using a 2 × 2 contingency x2 test of association and the difference expressed as percentage change. Power analysis based on a x2 distribution, was used to calculate the number of occupied squares needed during future surveys so as to demonstrate no significant decline from the current survey.

*2.2. Population*

The aim of this paper was to conduct a conservation assessment for the otter in Ireland that would fulfil the requirements of EC Habitat Directive; including an assessment of population change. We did not aim to either evaluate previous population assessment methods or develop new (and thus non-directly comparable) methods. A baseline population estimate was taken from the first national otter survey of Ireland during 1981–82 (Chapman and Chapman, 1982); back calculated by Ó Néill (2008) and peer-reviewed and published by Marnell et al. (2011). Baseline total abundance of adult breeding females was taken as 8300 [95%CI 7600–9800] individuals (Ó Néill, 2008; Marnell et al., 2011). Thus, we followed their methods as closely as possible to provide temporal comparability with the previous population estimate. Specifically, female otter abundance was estimated based on habitat- and productivity-specific density (individuals km−1) described in Table 1. The total length of riparian corridor (streams, rivers, lake edge) and coastline was calculated using a line vector shapefile and ArcGIS 10.1 (ESRI, California, USA). Streams were categorised as <2 m and 2–5 m whilst rivers were categorised as 5–10 m, 10–20 m, 20–40 m and >40 m in Northern Ireland where data were available from the Northern Ireland Environment Agency (shapefile dated 11/08/2006). Linear data for riparian length for the Republic of Ireland were obtained from the National Parks & Wildlife Service (shapefile dated 28/10/2008) but width had to be based on mean estimates from ground-truthed data gathered during previous otter surveys (Chapman and Chapman, 1982; Bailey and Rochford, 2006) where streams were estimated to be on aver- age 4.2 m wide and rivers 12.9 m wide. Typically, otters do not forage >80 m from river banks or lake or coastal shores (Kruuk and Moorhouse, 1991). Consequently, rivers >80 m wide were taken as representing two banks rather than one (as assumed for all rivers <80 m). Similarly, lake or coastal lines were mapped with a 80 m line length resolution, whereby edge habitats were treated as coincident when they were within 80 m of each other as they gave access to the same foraging habitat.

Streams, rivers, and lakes were further classified according to their trophic status, as defined by their levels of orthophosphate (low productivity = 0.00–0.02 mg l−1; intermediate productivity = 0.02–0.04 mg l−1 and high productivity >0.04 mg l−1).

Measurements of orthophosphate in water were derived from 2177 sites throughout Ireland from 2008 to 2010 collected by the Environmental Protection Agency in the Republic of Ireland and the Water Management Unit, Northern Ireland Environment Agency in Northern Ireland. Values were interpolated for areas with no measurements using the Kriging tool in Spatial Analyst for ArcGIS. Densities were subsequently adjusted according to altitude following the proportions identified by Ruiz-Olmo (1998). Coastal density was classified according to the underlying geology derived from the All Ireland Bedrock Map obtained from the Geological Survey of Ireland and was assumed to be independent of produc- tivity (Ó Néill, 2008). Mean density of otters (adult females.km2) was calculated per River Basin District and plotted using ArcGIS to demonstrate regional variation.

Breeding female otters have more stable home ranges than males or juveniles (Kruuk, 1995, 2006). Sex ratios of adult otter populations are rarely 1:1 (male:female), as might be expected for mammals at birth, but range widely e.g. 1:1 (Hung et al., 2004), 1.2:1 (Hájková et al., 2011), 1.3:1 (Philcox et al., 1999), 1.4:1 (Dallas et al., 2003 from carcasses), 1.5:1 (Dallas et al., 2003 from spraint) and 3.6:1 (Lanszki et al., 2009). Thus, estimating adult female abun- dance was deemed more reliable than estimating total abundance by multiplying female numbers by a factor of 2.

### *2.3. Habitat and future prospects*

Surveyor, rainfall in the week prior to survey and the number of bridges on each 600 m stretch of river have been shown to strongly negatively bias the Standard Otter Survey method (Reid et al., 2013a). Thus, each of these three variables was included in analysis of otter incidence. A further 28 variables were listed as perceived pressures extracted and modified from O'Sullivan (1996) and Foster-Turley et al. (1990) who listed major and specific threats to otters as recorded in 29 European countries/regions (Table 2). Only 9 such pressures were recorded at >10% of sites and retained for inclusion in analysis. A further 69 habitat (both aquatic and terrestrial) variables were recorded during the survey describing river size, flow regime, substrate, prey availability, bank type and management, vegetation, water use and adjacent landcover. Of these, 9 were deemed ecologically relevant enough for inclusion in analysis without modification (Table 3a), whilst the remaining 60 were reduced by a series of Principal Components Analyses (PCA) to 28 variables (Table 3b) yielding a final total of 37 candidate explanatory variables.

Variance in otter occurrence was examined within each habitat type (rivers, lakes and the coast) using Generalized Linear Models (GLMs) assuming a binomial error structure and a logit link function where otter occurrence (presence or absence) was fitted as the dependent variable. All candidate explanatory variables were tested for multicollinearity. One of each pair of significant bivariates ($r > 0.5$); the one with the weakest correlation coefficient with otter presence, was removed to ensure that all tolerance values were >0.1 and all variance inflation factor values were <10.0 (Quinn and Keough, 2002). To allow the direct comparison of regression coefficients, variables were standardised to have a $\bar{x} = 0$ and a a = 1 prior to analysis. All possible model permutations were created and ranked using AIC values. The Akaike weight ($\omega_i$) of each model was calculated within the top set of $N$ models, where the value of $\Delta AIC \leq 2$ units (Burnham and Anderson, 2002). The $\omega_i$ of each model is the relative likelihood of that model being the best within a set of $N$ models. To calculate the importance of each variable relative to all other variables, the $\Sigma\omega_i$ of all models within the top set of models that contained the variable of interest was calculated and the variables ranked by $\Sigma\omega_i$ (McAlpine et al., 2006); the larger the value of $\Sigma\omega_i$ (which varies between 0 and 1), the more important the variable. Multimodel inference and model averaging was used to determine the effect size (β coefficient) of each variable across the top set of models (Burnham and Anderson, 2002). Variables that had equal $\Sigma\omega_i$ values were ranked in order of the magnitude of their model averaged regression coefficients.

## 3. Results

Otters were widespread throughout Ireland during 1993–2006(Fig. 1a). Their baseline range in the Republic of Ireland was665 × 10 km cells and in Northern Ireland was 170 cells. Otters remained widespread during 2007–2011 (Fig. 1b). Their current range in the Republic of Ireland was 870 cells, representing a significant 52% increase (X2df=1= 11.3, $p < 0.001$) and in North-ern Ireland was 189 cells, representing a marginal 6% increase (X2df=1= 2.8, $p = 0.09$; Fig. 1c). As their current distribution was larger than that recorded at baseline, the range was revised and when reassessed on an All-Ireland scale all 1015 cells available for occupation were deemed suitable for the species. During 2007–2011, otters were recorded in 707 cells (70%) of their All-Ireland range. Power analysis suggested that the target for future surveys should be to record the otter as present in 160–182 cells in Northern Ireland, 504–581 cells in the Republic of Ireland or 666–746 cells throughout Ireland in order to demonstrate no significant change ($p < 0.05$) in its distribution.

Landscape productivity throughout Ireland changed between 1993–2006 and 2007–2011 with areas of low orthophosphate concentration expanding west-to-east replacing some areas of intermediate productivity, although eastern and southern areas of high productivity remained largely stable due to association with intensive agriculture (Fig. 2a and b). The density of suitable habitat for otters (streams, rivers and lake edge) was highest in the Western River Basin District whilst coastal complexity was high in both the Western and North Eastern River Basin Districts (Fig. 2c). The total estimate of adult otter abundance throughout Ireland (Table 4) was 9400 (95%CI 8700–12,200) breeding females. Whilst estimates of otter incidence at survey sites were largely uniform between River Basin Districts (Reid et al., 2013a), variance in the occurrence and density of suitable habitat resulted in regional variation in estimated otter density which was highest in the Western and North Eastern River Basin Districts (Fig. 2d).

The detection of otter tracks and signs was biased by surveyor and, at rivers, the cumulative volume of rain in the week prior to survey and the number of bridges present. At rivers, field signs of otters were positively associated with river size (Figs. 3a and 4), banks >1 m high sloping at $>30^\circ$, substrates composed of cobbles, gravel, boulders and exposed bedrock, channel and side bars and salmonid biomass (Fig. 3a). There were no significant variables retained in the top model of otter occurrence at lakes (Fig. 3b). Otter occurrence on the coast was positively associated with the biomass of salmonids in adjacent rivers running out to sea (Fig. 3c).

Some level of perceived disturbance (on an ordinal scale from 1 to 5) was recorded at 578 sites (59%) but 53% of these had a score $\leq 3$ (intermediate levels). Sources of disturbance included canal resectioning with bank maintenance at 216 sites (22%) and canalisation with mechanical weed control at 110 sites (11%). Boating activity and harbours occurred at 94 sites (10%) whilst angling, shooting and game keepering were present at 212 sites (22%). Mink were recorded at 117 sites out of 841 sites (14% occurrence). None of these perceived pressures or water quality were determined as actual threats as none were retained in the top models of otter occurrence, and thus had no discernible negative effect on otter occurrence in either rivers, lakes or the coast (Fig. 3).

## 4.0. Discussion

### *4.1. Range*

We demonstrated not only that there has been no decline in the distribution of the otter throughout Ireland, but that it's known range is more extensive than previously reported. Consequently, the range established during the baseline survey, against which future changes are supposed to be measured, was reassessed to include all possible 10 km squares available for occupation. All future surveys should thus be compared to the current study and not baseline data. As the species remains widespread the conservation assessment parameter of 'range' was judged favourable throughout Ireland and in both political jurisdictions.

### *4.2. Population*

Current estimates of species incidence for each River Basin District, corrected previously for negative survey bias by Reid et al. (2013a) were used to estimate the adult breeding female population during 2010–11 to be 9400 [95%CI 8700-12,200] individuals. As the 95% confidence intervals between the current and baseline estimated overlapped considerably we conclude that the current population was not significantly different from that estimated at baseline. The same was true for the Republic of Ireland taken separately (Table 4) but the population estimate for Northern Ireland increased significantly (i.e. the 95% confidence intervals did not overlap) from 1100 [95%CI 1000-1400] to 1600 [95%CI 1500-2000] adult females between 1981–82 and 2010–11. In any case, the conservation assessment parameter of 'population' was judged favourable in both jurisdictions and throughout the island as a whole. It is noteworthy that despite no variation in otter incidence being observed between regions after correction for survey bias (Reid et al., 2013a), variation in the availability of suitable habitat led to regional variation in predicted densities. Specifically, otter densities appeared highest in the Western and North Eastern River Basin Districts where the former had a high density of streams, rivers, lakes and other inland waterways whilst both districts possessed notably complex, convoluted coastlines providing greater habitat availability for otters and thus supported higher densities per unit area.

It should be noted that population estimates were based on radiotracking studies that dated from 1984 to 2008. Mammalian home range size can vary according to numerous local factors e.g. habitat characteristics, human impact and landscape connectivity, all of which can influence source-sink movements. Thus, caution must be used when generalising such data to a country scale. More- over, the size of home ranges reported from radiotelemetry studies can be influenced by methodological details which may also vary between studies e.g. the duration of tracking, such that generalising over time might be an additional source of bias or error.

*4.3. Habitat*

Variation in otter occurrence was more strongly influenced by bias in Standard Otter Survey method (surveyor ability and search effort, rainfall and the number of bridges on each stretch of river) than by landscape-scale parameters derived from a GIS- based approach (Reid et al., 2013a). Here, we demonstrate a broadly similar result using ground-truthed, habitat survey data. It is essential to account for known biases in surveyor methodologies when analysing habitat associations otherwise variation attributable to the survey method may be erroneously attributed to environmental parameters which may appear more important than they might otherwise be in reality. Having accounted for survey biases, otter incidence was positively associated with large (wide and deep) rivers with in-channel features such as bars and side bars, hard substrates including gravel, cobbles, boulders and exposed bedrock with high, moderately sloping banks. These findings are consistent with previous studies which have shown similar results (Bailey and Rochford, 2006).

Whilst these may represent true ecological relationships it may also be the case that surveyors preferentially examine bars and side bars or boulders and exposed bedrock for spraint. If the bank was tall and sloping it may have provided a better vantage point from which to search for spraint than if the bank was low and flat. Otters are known to be positively associated with river width and depth (Bailey and Rochford, 2006) with larger rivers supporting higher densities of individuals (Ó Néill, 2008) due to reduced competition of territorial space and food resources. However, large rivers tend to have few in-channel features, for example, bars and side bars or exposed boulders, whilst their depth often prohibits the survey of any such features present. Thus, possible sprainting sites on large rivers generally include bridge footings or boulders at the bank whilst these areas were usually the only places where thorough surveys could be completed. Thus, it might be that even the relationships found here may be confounded by survey bias rather than being meaningful ecological associations.

Otter occurrence at rivers was positively influenced by salmonid biomass, derived from electrofishing data in riffle habitat. Salmonids constitute, on average, 18–31% of the diet of otters in Ireland (Reid et al., 2013b) though in some catchments this can be as high as 81% (Fairley and Wilson, 1972). Thus, they are the single most important prey item in the diet of Irish otters. Nevertheless, it has been shown that they do not actively select salmonid over non-salmonid prey (Reid et al., 2013b). This may be because salmonids are present in practically every waterway in Ireland with most rivers containing brown trout (*Salmo trutta*) and many lakes stocked with rainbow trout (*Oncorhynchus mykiss*). Salmonid density varies throughout Ireland being highest in rivers in the north- and south-west that drain into the Atlantic Ocean suggesting that these areas are important in sustaining Atlantic salmon (*S.*

*salar*) and sea trout (*S. trutta* morpha *trutta*) returning to freshwater and augmenting resident salmonid numbers. Thus, otter abundance may reflect variation in salmonid abundance on a very coarse scale whilst at a regional level there is little correspondence between otter and salmonid populations. Otter occurrence on the coast has been shown to be positively influenced by the coverage of adjacent inland freshwater-dominated landscapes, principally high densi- ties of riparian corridors and standing freshwater (lakes) fringed with broad-leaved woodland (Reid et al., 2013a). Coastal otters are dependent on freshwater rivers for bathing to maintain their fur or for foraging (Kruuk, 2006; Chanin, 2013). Certainly, coastal radio-tracked otters in Co. Cork have been seen to travel upstream into freshwater systems on occasions (de Jongh et al., 2010). All coastal sites surveyed in the current study were adjacent to the mouth of a river or stream. Our results suggest that otters on the coast were positively influenced by salmonid biomass in these adja- cent rivers supporting the previous supposition that coastal otters may return to freshwater to feed. Thus, in coastal areas otters may indeed actively select salmonids, principally, sea trout and salmon.

None of the factors that significantly influenced otter occurrence (river size, substrate type, bank elevation and slope or salmonid biomass) are likely to be limiting or impacted detrimentally by human activities. Consequently, the conservation assessment parameter of ‘habitat’ was judged favourable throughout Ireland.

*4.4. Future prospects*

It was notable that otter occurrence was unaffected by water quality (Q-values), perceived levels of disturbance, mink occurrence or water use. Other pressures such as by-catch in fishing gear and road kill may be important locally but were not considered to be significant threats to the long-term persistence of the species regionally or at the national level, especially considering the widespread distribution of the species. Consequently, the conservation assessment parameter of ‘future prospects’ was judged favourable throughout Ireland.

## 5. Conclusions

The conservation status of the otter in the Republic of Ireland, Northern Ireland and throughout the island was judged Favourable. The previous Article 17 assessment from 1993 to 2006 for the Republic of Ireland (NPWS, 2008) was deemed Poor indicating an improving trend. However, this was considered to be due to improved knowledge and more accurate data rather than a real temporal trend (also see Reid et al., 2013a). We recommend that a formal trans-boundary assessment is submitted for the next Article 17 report and strongly advocate that neighbouring EU member states that share habitats and species across comparable ecoregions should work together to standardise and synchronise monitoring and surveillence regimes to ensure conservation assessment data are biologically meaningful.

## Acknowledgements

This study was part-funded by the National Parks & Wildlife Service (NPWS), Department of Arts, Heritage and the Gaeltacht (DAHG) covering the Republic of Ireland and part-funded by the Northern Ireland Environment Agency (NIEA) via the Natural Heritage Research Partnership (NHRP) with *Quercus*, Queen’s University Belfast (QUB). We also thank landowners and farmers throughout Ireland who allowed access to their land.

## References

92/43/EEC, 1992. Conservation of Natural Habitats and of Wild Fauna and Flora. http://ec.europa.eu/environment/nature/legislation/habitatsdirective Last accessed 09/10/2013.

Bailey, M., Rochford, J., 2006. Otter Survey of Ireland 2004/2005. Irish Wildlife Man- uals No. 23. National Parks and Wildlife Service, Department of Environment, Heritage and Local Government, Dublin.

Burnham, K.P., Anderson, D.R., 2002. Model Selection and Multimodel Inference: A Practical Information-Theoretic Approach, second ed. Springer-Verlag, New York.

Chanin, P., 2013. Otters. Whittet Books Ltd, Stansted.

Chapman, P.J., Chapman, L.L., 1982. Otter survey of Ireland 1980-81. The Vincent Wildlife Trust, London.

de Jongh, A., O'Neill, L., de Jongh, T., 2010. Coastal otters (Lutra lutra) in Roaringwater bay, Ireland. National Parks and Wildlife Service, Department of Environment, Heritage and Local Government, Dublin.

Dallas, J.F., Coxon, K.E., Sykes, T., Chanin, P.R.F., Marshall, F., Carss, D.N., Bacon, P.J., Piertney, S.B., Racey, P.A., 2003. Similar estimates of population genetic com- position and sex ratio derived from carcasses and faeces of Eurasian otter Lutra lutra. Mol. Ecol. 12, 275–282.

Durbin, L., 1996. Individual differences in spatial utilisation of a river-system by otters (Lutra lutra). Acta Theriol. 41, 137–147.

European Commission, 2006. Assessment, monitoring and reporting under Article 17 of the Habitats Directive: Explanatory Notes & Guidelines, http://www.artdata.slu.se/filer/gybs/notes guidelines report art17 final.pdf Last accessed 09/10/2013.

Evans, D., Arvela, M., 2011. Assessment and Reporting under the Habitats Directive. European Topic Centre on Biological Diversity, Paris, France http://bd.eionet.europa.eu/activities/Reporting/Article 17/reference portal

Fairley, J.S., Wilson, S.C., 1972. Autumn food of otters on the Agivey River, Co. Lon- donderry, Northern Ireland. J. Zool. 166, 468–469.

Foster-Turley, P., Macdonald, S.M., Mason, C.F., 1990. Otters an Action Plan for Conservation. International Union for the Conservation of Nature, Gland, Switzerland.

Green, J., Green, R., Jefferies, D.J., 1984. A radio-tracking survey of otters (Lutra lutra) on a Perthshire river system. Lutra 27, 85–145.

Gutleb, A.C., Kranz, A., 1998. Estimation of polychlorinated biphenyl (PCB) levels in livers of the otter (Lutra lutra) from concentrations in scats and fish. Water Air Soil Poll 106, 481–491.

Hájková, P., Zemanová, B., Roche, K., Hájek, B., 2011. Conservation genetics and non- invasive genetic sampling of Eurasian otters (Lutra lutra) in the Czech and Slovak republics. In:

Proceedings of Xth International Otter Colloquium, IUCN Otter Spec Group Bull. 28A, pp. 127–138.

Henle, K., Bauch, B., Auliya, M., Külvik, M., Pe'er, G., Schmeller, D.S., Framstad, E., 2013. Priorities for biodiversity monitoring in Europe: a review of supranational policies and a novel scheme for integrative prioritization. Ecol. Indic. 33, 5–18.

Hung, C.M., Li, S.H., Lee, L.L., 2004. Faecal DNA typing to determine the abundance and spatial organisation of otters (Lutra lutra) along two stream systems in Kinmen. Anim. Conserv. 7, 301–311.

Jefferies, D.J., 1986. The value of otter Lutra lutra surveying using spraints: An analysis of its success and problems in Britain. Otters 1, 25–32.

JNCC, 2007. Second Report by the UK under Article 17 on the implementation of the Habitats Directive from January 2001 to December 2006. Joint Nature Con- servation Committee, Peterborough, http://jncc.defra.gov.uk/page-4060 Last accessed 09/10/2013.

Kruuk, H., 1995. Wild Otters. Predation and populations. Oxford University Press, Oxford.

Kruuk, H., 2006. Otters, ecology, behaviour and conservation. Oxford University Press, Oxford.

Kruuk, H., Moorhouse, A., 1991. The spatial organization of otters (Lutra lutra) in Shetland. J. Zool. 224, 41–57.

Kruuk, H., Carss, D.N., Conroy, J.W., Durbin, L., 1993. Otter (Lutra lutra) numbers and fish productivity in rivers in north-east Scotland. Symp. Zool. Soc. Lond. 65, 171–191.

Lanszki, J., Orosz, E., Sugár, L., 2009. Metal levels in tissues of Eurasian otters (Lutra lutra) from Hungary: variation with sex, age, condition and location. Chemo- sphere 74, 741–743.

Lenton, E.J., Chanin, P.R.F., Jefferies, D.J., 1980. Otter survey of England 1977-79.
Nature Conservancy Council, London.

Macdonald, S.M., Mason, C.F., 1976. The status of the otter (Lutra lutra L.) in Norfolk.
Biol. Conserv. 9, 119–124.

Marnell, F., Ó Néill, L., Lynn, D., 2011. How to calculate range and population size for the otter? The Irish approach as a case study. IUCN Otter Spec. Group Bull. 28 (8), 15–22.

Mason, C.F., MacDonald, S.M., 1986. Otters. Ecology and Conservation. Cambridge University Press, New York.

Mason, C.F., Wren, C.D., 2001. Carnivora. In: Shore, R., Rattner, B. (Eds.), Ecotoxicology of Wild Mammals. Wiley, Chichester, pp. 315–370.

McAlpine, C., Rhodes, A.J.R., Callaghan, J., Bowen, M.E., Lunney, D., Mitchell, D.L., Pullar, D.V., Pssingham, H.P., 2006. The importance of forest area and configura- tion relative to local habitat factors for conserving forest mammals: a case study of koalas in Queensland, Australia. Biol. Conserv. 132, 152–165.

NPWS, 2008. The status of EU protected habitats and species in Ireland. National Parks and Wildlife Service, Dublin, Ireland.

Ó Néill, L., 2008. Population dynamics of the Eurasian otter in Ireland. Integrating density and demography into conservation planning. Unpublished PhD thesis. Trinity College, Dublin.

O'Sullivan, W.M., 1996. Otter conservation: factors affecting survival, with particular reference to drainage and pollution within an Irish river system. In: Reynolds, J.D. (Ed.), The Conservation of Aquatic Systems. Royal Irish Academy, Dublin, pp. 117–133.

Parry, G.S., Bodger, O., McDonald, R.A., Forman, D.W., 2013. A systematic re-sampling approach to assess the probability of detecting otters Lutra lutra using spraint surveys on small lowland rivers. Ecol. Inform. 14, 64–70.

Pereira, H.M., Cooper, D., 2006. Towards the global monitoring of biodiversity loss. Trends Ecol. Evol. 21, 123–129.

Philcox, C.K., Grogan, A.L., Macdonald, D.W., 1999. Patterns of otter (Lutra lutra) road mortality in Britain. J. Appl. Ecol. 36, 748–762.

Preston, J., Prodohl, P., Portig, A., Montgomery, W.I., 2006. Reassessing Otter Lutra lutra distribution in Northern Ireland. Environment and Heritage Service Research and Development Series. No. 06/24. Belfast.

Quinn, G.P., Keough, M.J., 2002. Experimental Design and Data Analysis for Biologists. Cambridge University Press.

Reid, N., Lundy, M.G., Hayden, B., Lynn, D., Marnell, F., McDonald, R.A., Montgomery, W.I., 2013a. Detecting detectability: identifying and correcting bias in binary wildlife surveys demonstrates their potential impact on conservation assess- ments. Eur. J. Wildlife Res. 59 (6), 869–879.

Reid, N., Thompson, D., Hayden, B., Marnell, F., Montgomery, W.I., 2013b. Review and quantitative meta-analysis suggests of diet suggests the Eurasian otter (Lutra lutra) is likely to be a poor bioindicator. Ecol. Indic. 26, 5–13.

Ruiz-Olmo, J., 1998. Influence of altitude on the distribution, abundance and ecology of the otter (Lutra lutra). Symp. Zool. Soc. Lond. 71, 159–176.

Ruiz-Olmo, J., Loy, A., Cianfrani, C., Yoxon, P., Yoxon, G., de Silva, P.K., Roos, A., Bisther, M., Hajkova, P., Zemanova, B., 2008. Lutra lutr. In: IUCN 2013. IUCN Red List of Threatened Species. Version 2013. 1, http://www. iucnredlist.org/details/12419/0 Last accessed 09/10/2013.

Schmeller, D.S., 2008. European species and habitat monitoring: where are we now? Biodivers. Conserv. 17 (4), 3321–3326.

West, R.B., 1975. The Suffolk otter survey. Suffolk Natural History 16, 378–388. Yoxon, P., 1999. Geology and otter distribution on Skye. Dissertation. Open University, Milton Keynes, UK.

**Table 1**
Adult female otter density (individuals $\text{km}^{-1}$) with 95% confidence intervals in parentheses for various habitats throughout Ireland at three levels of landscape productivity.

| Country (source) | Habitat category | Productivity (orthophosphate) | | |
|---|---|---|---|---|
| | | Low[a] (0.00–0.02 $\text{mg l}^{-1}$) | Intermediate[c] (0.02–0.04 $\text{mg l}^{-1}$) | High[b] (>0.04 $\text{mg l}^{-1}$) |
| Republic of Ireland (OSI) | Streams 4.2 m | 0.05 [0.05–0.06] | 0.06 [0.05–0.10] | 0.07 [0.05–0.10] |
| | Rivers 12.9 m | 0.05 [0.05–0.06] | 0.08 [0.05–0.14] | 0.12 [0.11–0.14] |
| | Lakes | 0.05 [0.05–0.06] | 0.09 [0.05–0.21] | 0.17 [0.15–0.21] |
| Northern Ireland (OSNI) | Streams <2 m | 0.05 [0.05–0.05] | 0.05 [0.05–0.05] | 0.05 [0.05–0.05] |
| | Streams 2–5 m | 0.06 [0.05–0.08] | 0.07 [0.05–0.10] | 0.08 [0.05–0.12] |
| | Rivers 5–10 m | 0.08 [0.05–0.12] | 0.09 [0.07–0.12] | 0.10 [0.08–0.14] |
| | Rivers 10–20 m | 0.10 [0.08–0.14] | 0.11 [0.09–0.14] | 0.12 [0.11–0.15] |
| | Rivers 20–40 m | 0.12 [0.11–0.15] | 0.13 [0.11–0.15] | 0.13 [0.11–0.16] |
| | Rivers >40 m | 0.13 [0.11–0.16] | 0.14 [0.12–0.16] | 0.15 [0.13–0.17] |
| | Lakes | 0.05 [0.05–0.06] | 0.09 [0.05–0.21] | 0.17 [0.15–0.21] |
| Coastlines (Geological Survey of Ireland) | Paleozoic[d] | | 0.43 [0.38–0.49] | |
| | Carboniferous[e] | | 0.43 [0.38–0.49] | |
| | Devonian[f] | | 0.10 [0.09–0.11] | |
| | Igneous[d] | | 0.10 [0.09–0.11] | |
| | Mesozoic[d] | | 0.66 [0.58–0.75] | |
| | Pre-Cambrian[d] | | 0.18 [0.16–0.20] | |

Density estimates were derived from [a]Green et al. (1984), Kruuk et al. (1993), Durbin (1996), Kruuk (2006); [b]O'Néill (2008); [c]Intermediate between low and high trophic status with most extreme confidence intervals; [d]Yoxon (1999); [e]Carboniferous limestone was assumed to be similar to Cambrian rock (Yoxon, 1999) and [f]Otter density on Devonian rock was assumed similar to that on Igneous rock (Kruuk, 1995; Yoxon, 1999; H. Kruuk *pers. comms.*).
Note the effect of geology was assumed independent of productivity (Ó Néill, 2008).

**Table 2**
List of perceived pressures on otter populations extracted and modified from O'Sullivan (1996) and Foster-Turley (1990) who listed major and specific threats to otters as recorded in 29 European countries/regions. These have been translated into their corresponding Habitats & Species Directive threat codes.

| Type | Name | EU pressure & threat | |
|---|---|---|---|
| | | Code | Description |
| Adjacent land use | Farm livestock | A04 | Grazing |
| | Arable | A02.01 | Agricultural intensification |
| Water use | Abstraction | J02.06 | Water abstractions from surface waters |
| | Wetland drainage | J02.07.01 | Groundwater abstractions for agriculture |
| | Boating* | G01.01 | Nautical sports (motorised and non-motorised) |
| | Bank angling* | F02.03 | Leisure fishing |
| | Shooting* | F05.05 | Shooting |
| | Game keepering* | F06.01 | Game/bird breeding station |
| | Aquaculture/fisheries | F01 | Marine or freshwater aquaculture |
| | Fyke netting | F02.01.02 | Netting |
| | Illegal killing | F03.01 | Hunting |
| | Hydroelectric scheme | J02.05.05 | Small hydropower projects, weirs |
| Weed control | Mechanical* | A10.01 | Removal of hedges and copses or scrub |
| | Chemical | H01.09 | Diffuse pollution to surface waters (not listed) |
| Bank management | Canalised* | J02.03.02 | Canalisation |
| | Resectioned* | J02.05.02 | Modifying structures of inland water courses |
| Pollution | Agricultural | H01.05 | Diffuse pollution to surface waters (agricultural/forestry) |
| | Domestic | H01.08 | Diffuse pollution to surface waters (household) |
| | Industrial | H01.01 | Pollution to surface waters by industrial plants |
| | Oil spillages | H03.01 | Oil spills in the sea |
| Construction | Piers | D03.01.02 | Piers/tourist harbours or recreational piers |
| | Moorings* | D03.01.03 | Fishing harbours |
| | Slipways | D03.01.01 | Slipways |
| | Fishing stands | E04 | Structures, buildings in the landscape |
| | Road traffic | D01.02 | Roads & motorways |
| | Development | E01 | Urbanisation areas, human habitation |
| Invasive species | American mink* | I01 | Invasive non-native species |
| | Giant Hogweed | I01 | Invasive non-native species |

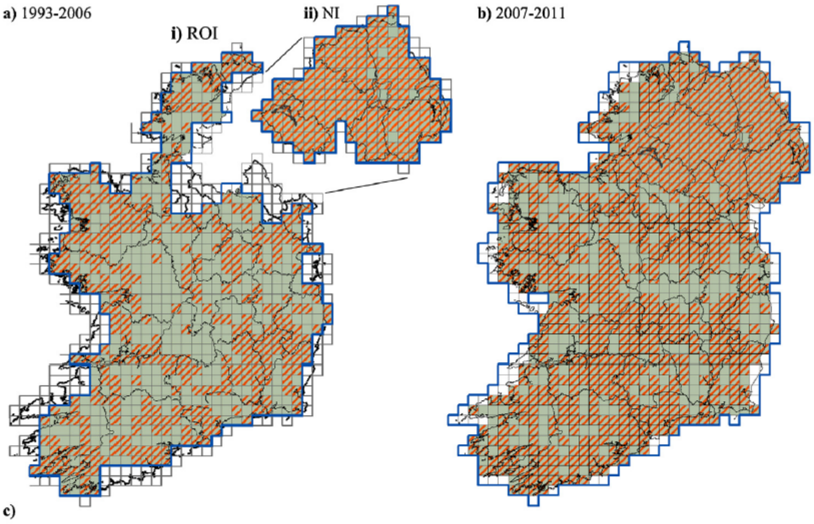
* highlights those pressures that occurred at >10% of sites ($n = 9$) which were included in otter occurrence modelling with a general measure of perceived site disturbance (an ordinal scale from 1 to 5).

**Table 3**
Explanatory variables selected for inclusion in models of otter occurrence. a) Nine variables were retained without modification; however, b) the remaining 60 were reduced to 28 variables using Principal Components Analysis (PCA) yielding at total of 37 candidate explanatory variables.

| Variable type | Explanatory variable | Description |
|---|---|---|
| **a) Unmodified variables** | | |
| Survey bias | Surveyor | There were a total of 17 survey teams throughout Ireland. Sixteen consisted of 75 conservation rangers from the National Parks & Wildlife Service (NPWS), Department of Arts, Heritage and the Gaeltacht, Republic of Ireland (ROI) whilst the sites in Northern Ireland (NI) were covered by one team consisting of two ecologists from Queen's University Belfast. |
| | Rainfall | Cumulative volume of rainfall (mm) during the 7 days prior to the survey extracted from the geographically closest weather station, namely, Aldergrove and Armagh Observatory (NI) or Met Éireann stations at Belmullet, Casement, Claremorris, Cork Airport, Dublin Airport, Malin Head, Shannon Airport and Valentia Observatory (ROI). |
| | No. of bridges | Number of bridges on each 600 m survey stretch of river or 300 m radius of the survey points at lakes and on the coast determined as the number of intersections between a river, stream, canal GIS shapefile and road shapefile. |
| Water quality | Q-values | Ecological Quality Ratings or Q-values measured at the closest Environmental Protection Agency (EPA) monitoring site to each otter survey site. The last period during which measurements were available varied from 2004 to 2010 but were typically recent. See http://www.epa.ie |
| Mink | Mink occurrence | Presence or absence of mink scat at each otter survey site |
| Disturbance | Disturbance | Categorical 6-level factor for perceived disturbance ranging from no disturbance present (0) to high levels of disturbance present (5) |
| | Livestock | Presence or absence of domestic stock that had access to the river bank |
| Prey availability | Salmonid biomass | Biomass of salmonid species including brown trout (*Salmo trutta*), rainbow trout (*Oncorhynchus mykiss*), sea trout (*S. trutta* morpha *trutta*) and Atlantic salmon (*Salmo salar*) interpolated for all sites using the Kriging tool in Spatial Analyst of ArcGIS derived from 77 locations from which electofishing data were available from Inland Fisheries Ireland. See http://www.fisheriesireland.ie |
| Tidal state (coastal sites only) | Tide | Categorical 3-level factor including low, intermediate and high tidal states. |
| **b) Variables derived from Principal Components Analysis of data collected in the field** | | |
| River size PCA | River size | Principal Component (PC) Axis 1 accounted for 72.3% of variance in river size (eigenvalue = 1.446) and was positively correlated with channel width ($r$ = +0.850) and channel depth ($r$ = +0.850) |
| Flow regime PCA | Slow flowing water | PC1 accounted for 26.5% of variance in flow regime (eigenvalue = 1.327) and was positively correlated with slow flowing water ($r$ = +0.828) and negatively correlated with fast flowing water ($r$ = –0.796) |
| | Fast flowing water | PC2 accounted for 21.9% of variance in flow regime (eigenvalue = 1.093) and was positively correlated with rapidly flowing water ($r$ = +0.951) |
| Substrate PCA | Cobble & gravel substrate | PC1 accounted for 21.5% of variance in substrate (eigenvalue = 1.293) and was positively correlated with cobbles ($r$ = +0.751) and gravel ($r$ = +0.640) |
| | Exposed bedrock & boulders | PC2 accounted for 20.4% of variance in substrate (eigenvalue = 1.224) and was positively correlated with exposed bedrock ($r$ = +0.630) and boulders ($r$ = +0.585) |
| | Sandy substrate | PC3 accounted for 17.4% of variance in bankside vegetation (eigenvalue = 1.045) and was positively correlated with sand ($r$ = +0.935) |
| Channel feature PCA | Channel & side bars | PC1 accounted for 44.7% of variance in channel features (eigenvalue = 1.341) and was positively correlated with in-channel bars ($r$ = +0.795) and side bars ($r$ = +0.701) |
| Aquatic vegetation PCA | Aquatic plants | PC1 accounted for 35.1% of variance in aquatic vegetation (eigenvalue = 1.405) and was positively correlated with submerged plants ($r$ = +0.718) and emergent vegetation ($r$ = +0.767) |
| Bankside vegetation PCA | Trees providing shade | PC1 accounted for 35.6% of variance in bankside vegetation (eigenvalue = 3.207) and was positively correlated with trees ($r$ = +0.766) including hawthorn ($r$ = +0.610), sycamore ($r$ = +0.618) and ash ($r$ = +0.686) with overhanding boughs ($r$ = +0.788) providing shade ($r$ = +0.716) |
| | Tall herbaceous plants & shrubs | PC2 accounted for 16.1% of variance in bankside vegetation (Eigenvalue = 1.446) and was positively correlated with tall herbs ($r$ = +0.824) and shrubs ($r$ = +0.708) |
| Bank type PCA | Low shallow banks | PC1 accounted for 22.8% of variance in bank type (eigenvalue = 1.596) and was positively correlated with banks <1 m high ($r$ = +0.837) with slopes <30° ($r$ = +0.741) |
| | Moderately high sloping banks | PC2 accounted for 18.1% of variance in bank type (eigenvalue = 1.266) and was positively correlated with banks 1–2 m high ($r$ = +0.690) with slopes 30–60° ($r$ = +0.656) |
| | High banks | PC3 accounted for 16.1% of variance in bank type (eigenvalue = 1.128) and was positively correlated with banks 2–3 m high ($r$ = +0.703) which did not slope steeply i.e. 60–90° ($r$ = –0.518) |
| Bank management PCA | Wild unmaintained banks | PC1 accounted for 17.9% of variance in bank management (eigenvalue = 1.434) and was positively correlated with wild banks ($r$ = +0.729) with no management ($r$ = +0.885) |
| | Chemical control of giant hog weed | PC2 accounted for 17.1% of variance in bank management (eigenvalue = 1.364) and was positively correlated with giant hogweed ($r$ = +0.777) and chemical control ($r$ = +0.775) |
| | Canalisation with mechanical weed control | PC3 accounted for 15.2% of variance in bank management (eigenvalue = 1.218) and was positively correlated with canalisation ($r$ = +0.916) and mechanical weed control ($r$ = +0.560) |
| | Resectioned and maintained | PC4 accounted for 15.0% of variance in bank management (eigenvalue = 1.201) and was positively correlated with resectioned ($r$ = +0.792) and maintained banks ($r$ = +0.509) |
| Water use PCA | Boating and harbours | PC1 accounted for 24.9% of variance in water use (eigenvalue = 1.181) and was positively correlated with boating activity ($r$ = +0.775) and harbours or moorings ($r$ = +0.837) |
| | Hunting activities | PC2 accounted for 23.7% of variance in water use (eigenvalue = 1.419) and was positively correlated with game keepering ($r$ = +0.709), shooting ($r$ = +0.562) and angling ($r$ = +0.577) |

Table 3 (*Continued*)

| Variable type | Explanatory variable | Description |
|---|---|---|
| Adjacent landcover PCA | Urban areas | PC1 accounted for 26.8% of variance in landcover (eigenvalue = 1.343) and was positively correlated with urban ($r$ = +0.752) and parks ($r$ = +0.803) |
| | Woodland | PC2 accounted for 25.5% of variance in landcover (eigenvalue = 1.276) and was positively correlated with broad-leaved woodland ($r$ = +0.675) and coniferous plantations ($r$ = +0.807) |
| Fish eating birds PCA | Fish eating birds | PC1 accounted for 52.2% of variance in fish eating bird presence (eigenvalue = 1.566) and was positively correlated with cormorants ($r$ = +0.822), gulls ($r$ = +0.734) and herons ($r$ = +0.592) |
| Shoreline PCA (lakes & coasts only) | Low shallow shores | PC1 accounted for 28.9% of variance in shoreline type (eigenvalue = 1.733) and was positively correlated with shorelines <5 m high ($r$ = +0.931) which sloped gently at 0–30° ($r$ = +0.907) |
| | Intermediately high sloping shores | PC2 accounted for 23.2% of variance in shoreline type (eigenvalue = 1.394) and was positively correlated with shorelines 5–20 m high ($r$ = +0.801) which sloped at 30–60° ($r$ = +0.844) |
| | High steep shores | PC3 accounted for 21.4% of variance in shoreline type (eigenvalue = 1.284) and was positively correlated with shorelines >20 m high ($r$ = +0.745) which sloped steeply at 60–90° ($r$ = +0.829) |
| Coastal habitat PCA (coasts only) | Rocky shores | PC1 accounted for 26.1% of variance in coastal habitat (eigenvalue = 1.321) and was positively correlated with rocky shores ($r$ = +0.740) and cliffs ($r$ = +0.691) |
| | Beeches & saltmarsh | PC2 accounted for 25.2% of variance in coastal habitat (eigenvalue = 1.242) and was positively correlated with saltmarsh ($r$ = +0.750) and negative correlated with beeches ($r$ = –0.771) |
| Shellfish PCA (coasts only) | Shellfish | PC1 accounted for 59.5% of variance in shellfish and mollusc presence (eigenvalue = 1.786) and was positively correlated with shellfish ($r$ = +0.905) and crabs ($r$ = +0.885) |




c)

| | | Northern Ireland | Republic of Ireland | All-Ireland |
|---|---|---|---|---|
| **Favourable reference range** | Baseline | 170 | 665 | 826 |
| (bold blue line) | Updated | 189 (+*11%*) | 870 (+*31%*) | **1,015** (+*23%*) |
| **Distribution** | Past | 163 | 358 | 515 |
| (hatched orange cells) | Current | 173 (+*6%*) | 543 (+*52%*) | **707** (+*37%*) |

**Fig. 1.** (a) The recorded '*distribution*' i.e. occupied 10 km cells (orange hatching), '*range*' i.e. cells enclosed by the observed distribution (green) and '*favourable reference range*' i.e. maximum likely extent of the range (bold blue line) for the otter in (i) Republic of Ireland (ROI) and (ii) Northern Ireland (NI) from 1993 to 2006 as reported previously under Article 17 of the EU Habitats Directive. Despite sharing 44 × 10 km cells along their common border both jurisdictions report to the European Commission separately meaning that the same cell can be otter positive in one country but otter negative in the other (hence the two maps). (b) The same representation for All-Ireland from 2007 to 2011 treated as a single biogeographical ecoregion. (c) A descriptive breakdown and analysis of temporal change allowing each jurisdiction to report separately. Note that the All-Ireland figures are not a summation of both countries due to shared cells along their border.

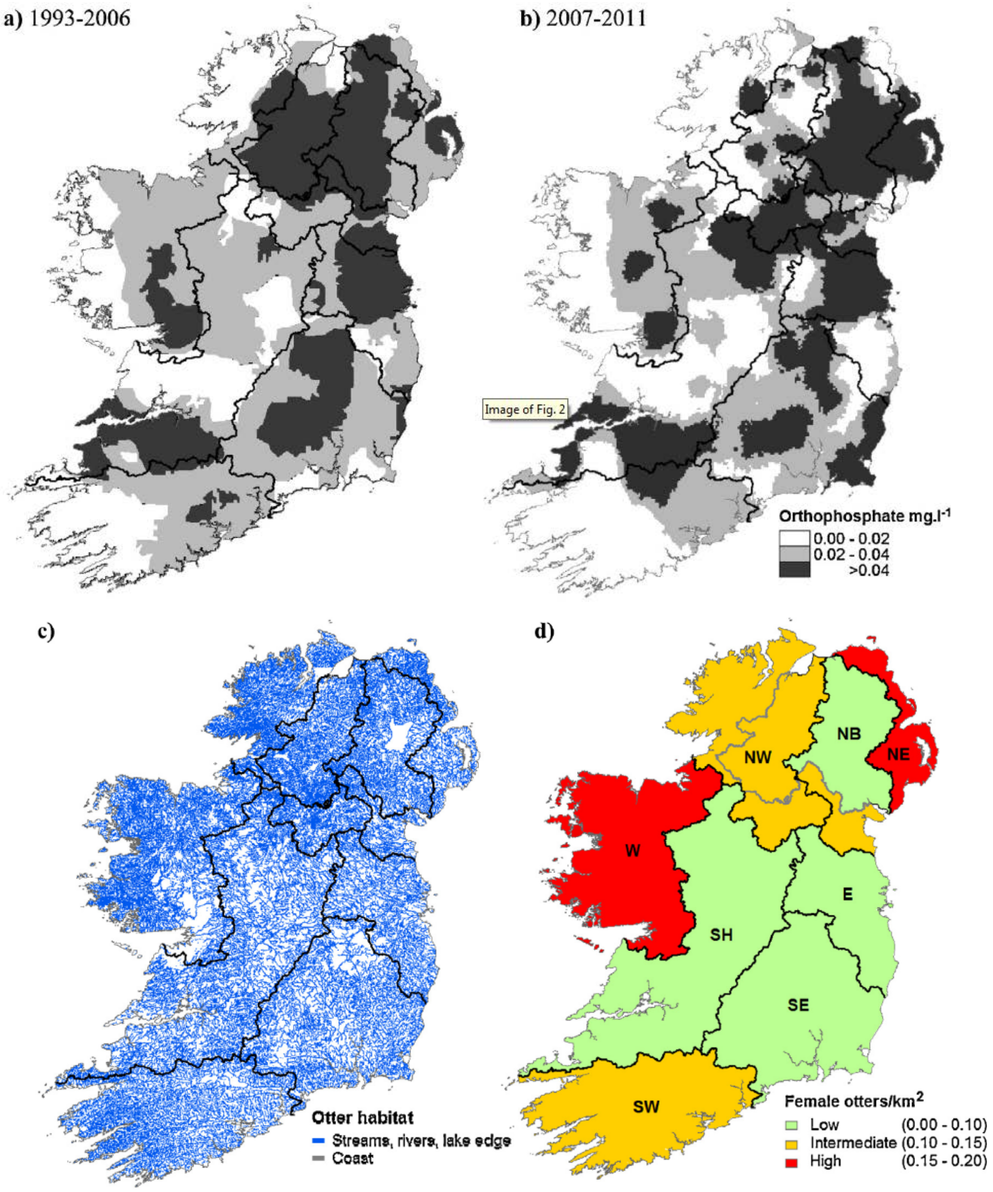



**Fig. 2.** Isoclines of orthophosphate levels (productivity) in Irish freshwaters during (a) 1993–2006 [extracted from Ó Néill, 2008] and (b) 2007–2011. Regional variation in (c) the density of suitable otter habitat represented by streams, rivers, lake edge and the coast and (d) estimated otter density. Bold black lines represent the boundaries of EU River Basin Districts including North Eastern (NE), Eastern (E), South Eastern (SE), Neagh-Bann (NB), Shannon (SH), North Western (NW), Western (W) and South Western (SW). The Northern Ireland border is presented by a dark grey bold line.

**Table 4**
Estimates of otter incidence during 1981–82 (extracted from Ó Néill, 2008 based on species incidence from Chapman and Chapman, 1982) compared to current estimates based on incidence from Reid et al. (2013a) given as adult females (individuals) ± 95% confidence intervals in parentheses.

| Country | River Basin | Population estimate | |
|---|---|---|---|
| | District | 1981–82 | 2010–11 |
| Republic of Ireland | Eastern | 552 [497–684] | 585 [556–742] |
| | Neagh Bann | 121 [107–153] | 223 [206–274] |
| | North Western | 927 [850–1106] | 1069 [1015–1316] |
| | Shannon | 1515 [1401–1779] | 1644 [1531–2200] |
| | South Eastern | 1024 [918–1295] | 1153 [1081–1593] |
| | South Western | 1204 [1121–1384] | 1311 [1158–1660] |
| | Western | 1784 [1664–2073] | 1809 [1671–2401] |
| | Sub-total | 7127 [6558–8474] | 7794 [7218–10,186] |
| Northern Ireland | Neagh Bann | 434 [407–514] | 555 [507–691] |
| | North Eastern | 231 [207–285] | 572 [518–679] |
| | North Western | 469 [435–554] | 510 [472–663] |
| | Sub-total | 1134 [1049–1353] | 1637 [1497–2033] |
| All-Ireland | TOTAL | 8261 [7607–9827][a] | 9431 [8715–12,219] |

[a] Note that these figures differ from those reported by Ó Néill (2008) due to the correction of a minor totalling error in the original calculations.

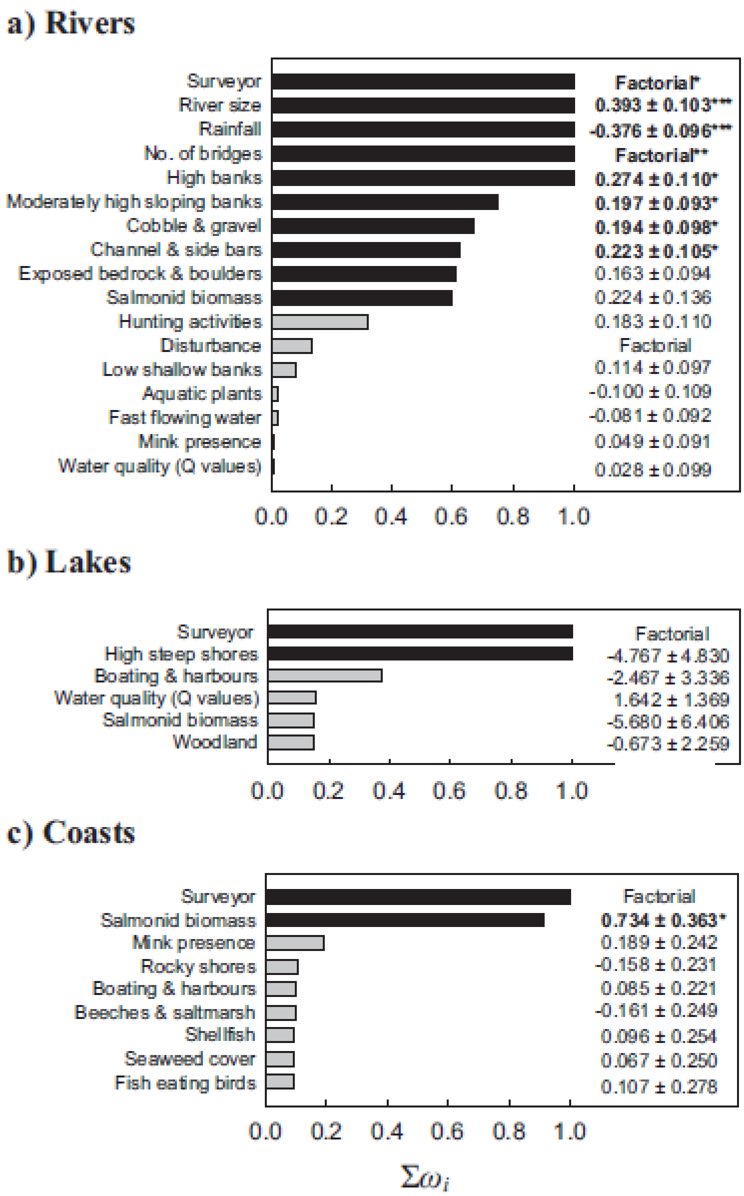



**Fig. 3.** Relative importance of explanatory variables in explaining variation in otter occurrence at a) rivers, b) lakes and c) the coast. Variables are ranked in order of the sum of their Akaike weights ($\Sigma\omega_i$) within the top set of models i.e. models with $\Delta AIC \leq 2$. Black bars indicate those variables that were retained in the best single approximating model (i.e. that with the lowest AIC value) and grey bars indicate variables included in all other models within the top set. Variables listed in Table 2 that are missing indicate that they were not included in the top set. Model averaged $\beta$ coefficients ± 95% confidence intervals for each covariate are shown to the right of each bar. Statistical significance is indicated in bold where $^{***}p < 0.001$, $^{**}p < 0.01$, $^{*}p < 0.05$.

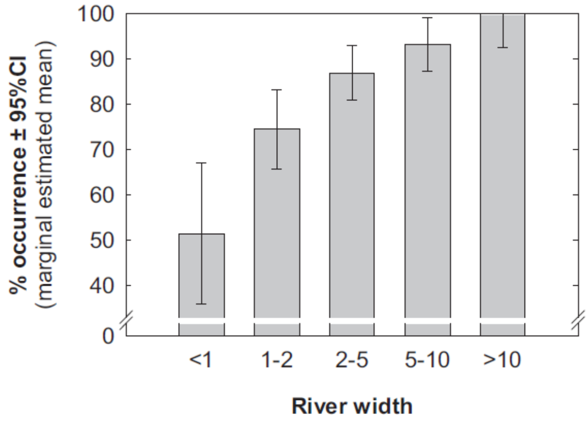



**Fig. 4.** Otter incidence (corrected for bias in surveyor, rainfall and the number of bridges) at rivers of varying size. Note that river width and depth were positively correlated ($r_p = 0.484$, $p < 0.001$).